MODELLING THE SEQUESTRATION POTENTIAL OF URBAN CARBON SINKS: A CASE STUDY OF FAIRFAX COUNTY, VA

by
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Executive Summary

This capstone project presents a model for assessing the potential carbon removal a city can achieve by enhancing the sequestration potential of its direct and embodied carbon sinks. Applying the model to Fairfax County, Virginia, this study finds that varying the city's policy options for managing its carbon sinks can have a small but measurable impact on its net carbon sequestration. Further, the study finds that maximizing the city's carbon sequestration requires changing its policies as the city hits specific thresholds of renewable energy usage. At these thresholds, maintaining extant policies decreases the city's net carbon sequestration: the city's former carbon offsets become net carbon sources. This study demonstrates that the model can both identify the timeframe in which such a transition is expected to occur, and test new policy options to identify those that will maximize the city's carbon sequestration at each stage of its transition to renewable energy. Modeling policy options in this way enables a city to plan its policy changes to achieve carbon neutrality more efficiently, or even explore the possibility of turning a city from a net carbon source to a net carbon sink in the future.

This study builds on work by Mohareb & Kennedy (2012), who suggested that cities may be able to increase their carbon sequestration potential by changing their policies for managing direct and embodied carbon sinks. This study confirms that cities can change their carbon sequestration potential via policy changes, based on analysis of a subset of potential urban carbon sinks, and that a city's policy choices may even determine whether the city acts as a long-term carbon source or sink for its embodied carbon. The model developed in this study could be expanded to include additional potential urban carbon sinks, and applied to other cities to inform their climate change mitigation plans.

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Introduction

There is growing evidence that cities can sequester significant carbon, both directly in urban land and vegetation, and indirectly through management of carbon embodied in city waste streams. In a U.K.-based study, Davies, Edmondson, Heinemeyer, Leake, & Gaston (2011) found that the city of Leicester could increase the carbon storage in its vegetation by 12% through replacing 10% of publicly owned grassland with tree cover. A study in Toronto, Canada that accounted for the gross carbon stored both in city vegetation and in waste products containing embodied carbon found that these sinks sequestered 2% of the city's emissions annually, but suggested that the city could increase this value through policy changes affecting the management of urban carbon sinks (Mohareb & Kennedy, 2012). Tozer & Klenk (2019) note that the spectrum of current carbon emissions reduction measures cannot enable a city to achieve full carbon neutrality, nor end its dependence on fossil fuels; further, J. Hansen et al. (2017) has noted that achieving current global warming targets requires removing carbon from the atmosphere. These studies point to the importance of modeling the sequestration potential of citywide carbon sinks to provide a tool by which policymakers can increase cities' carbon sequestration potential.

The study in Toronto by Mohareb & Kennedy (2012) appears to be the only previous work that has calculated carbon sequestration in both direct and embodied carbon sinks across a city. That study focused on developing methodologies for quantifying such sinks; therefore, it focused on gross carbon sequestration and did not account for emissions from management activities necessary to enhance the sinks. The current study focuses on net carbon sequestration to enable city leaders to account for the full impacts of policies affecting urban carbon sinks.

The current study also builds upon studies of individual carbon sinks. For direct carbon sinks, Nowak & Crane (2002) found that urban trees in the U.S. sequester 22.8 million metric tons of carbon per year. This echoes the conclusion of Davies et al. (2011), who found that urban trees sequester significant carbon. Davies et al. (2011) also found that herbaceous vegetation in Leicester (such as parks, gardens, roadsides, stream banks, school fields, and golf courses) sequestered the equivalent of the average emissions of 50,000 cars annually. Both of these studies accounted for carbon emissions from deadwood, which Orozco-Aguilar et al. (2018) found reduced carbon sequestration by a non-negligible amount. Such results suggest that net carbon sequestration by urban trees and herbaceous vegetation may be significant.

Previous studies have also found that carbon embodied in food scraps, yard waste, and wood waste from construction activity has the potential to become an urban carbon sink through landfilling, composting, recycling, or burning waste in place of fossil fuels. Mohareb and Kennedy (2012) found that embodied carbon in waste products sent to landfills constituted the largest carbon stock in Toronto. Barlaz (1998) found that landfill practices that maintain anaerobic conditions prevent waste products from fully breaking down, sequestering their carbon content indefinitely. The U.S. EPA (2006) confirmed that embodied carbon waste becomes a carbon sink in a landfill that captures 70% of gases (e.g., methane) released during anaerobic decomposition and maintains an anaerobic environment of less than 10% oxygen. Brown's (2016) review of literature on this topic suggests that such results should hold true regardless of local climate. These results suggest that city landfills have the potential to act as carbon sinks.

Similarly, the EPA suggests that composting embodied carbon waste and applying it in place of synthetic fertilizers can reduce fossil fuel emissions (U.S. EPA, Office of Resource Conservation and Recovery & ICF International, 2019d). Brown (2016) confirmed that synthetic fertilizers are energy-intensive to manufacture, resulting in carbon emissions from burning fossil fuels. Further, Brown et al. (2011) found that compost applications can result in increased carbon sequestration in soil. These studies suggest that composting embodied carbon waste products has the potential to create an urban carbon sink both by increasing soil carbon and by decreasing fossil fuel emissions, but that the impact of this carbon sink will decrease over time if manufacturing processes become increasingly powered by renewable energy.

Finally, past research on recycling and burning wood waste suggests that both options have the potential to create sizeable urban carbon sinks. A study of wood waste in several industrialized countries found that such waste stored carbon equivalent to 2% of these countries' 1990 baseline emissions (Hashimoto, Nose, Obara, & Moriguchi, 2002). Mohareb and Kennedy (2012) found that Toronto's waste lumber alone sequestered as much carbon as 28% of the city's urban forest in an average year. Mead (2005) found that cities worldwide were converting only about 15% of wood waste to energy as of 1993, and noted that this percentage was lower for developed countries. This suggests that cities may contain a large amount of embodied carbon in wood waste. A study comparing the impacts of disposal methods for urban tree trimmings and construction wood waste in Michigan suggested that burying these materials for long-term carbon storage would lead to more carbon emissions reductions than burning the materials as biofuel (MacFarlane, 2009). A full lifecycle assessment of wood waste concluded that either recycling it or using it to replace coal as a heat source resulted in net carbon sequestration, but that using waste wood to generate electricity or replace heat sourced from natural gas resulted in net carbon emissions (Morris, 2017). These studies suggest the importance of comparing the carbon sequestration potential from wood waste across a city's waste management options.

The current study develops a model of major carbon sinks in cities to address the question: *What is the potential carbon removal a city can achieve by enhancing the sequestration potential of its direct and embodied carbon sinks?* The term "carbon sequestration" is used in this study to refer to greenhouse gases generally; the model converts emissions of all greenhouse gases to carbon dioxide equivalent emissions. The model is especially intended for suburban cities, which are numerous in the American urban landscape but may not have the budgets to undertake their own full studies of carbon sinks within their jurisdictions. As a case study, this paper applies the model to Fairfax County, Virginia, to demonstrate how the model may be used to identify management practices that maximize carbon sequestration in urban sinks under current and future energy source regimes. Based on the results of previous studies on this topic, it is expected that this study will find that the model can be used to identify policies that increase carbon sequestration over time in a particular city's direct and embodied sinks.

Methods

The model developed for this study was built using STELLA Professional, a software program designed for building dynamic models of complex systems. The model built for this study consists of individual modules that each calculate the net carbon sequestration of one type of direct or embodied carbon sink. The model also aggregates the output from all of the modules to produce an overall estimate of carbon sequestration in the modeled city over time. Figure 1 shows the main modules in the model, as they appear in STELLA. Screenshots of each module appear in the Appendix.

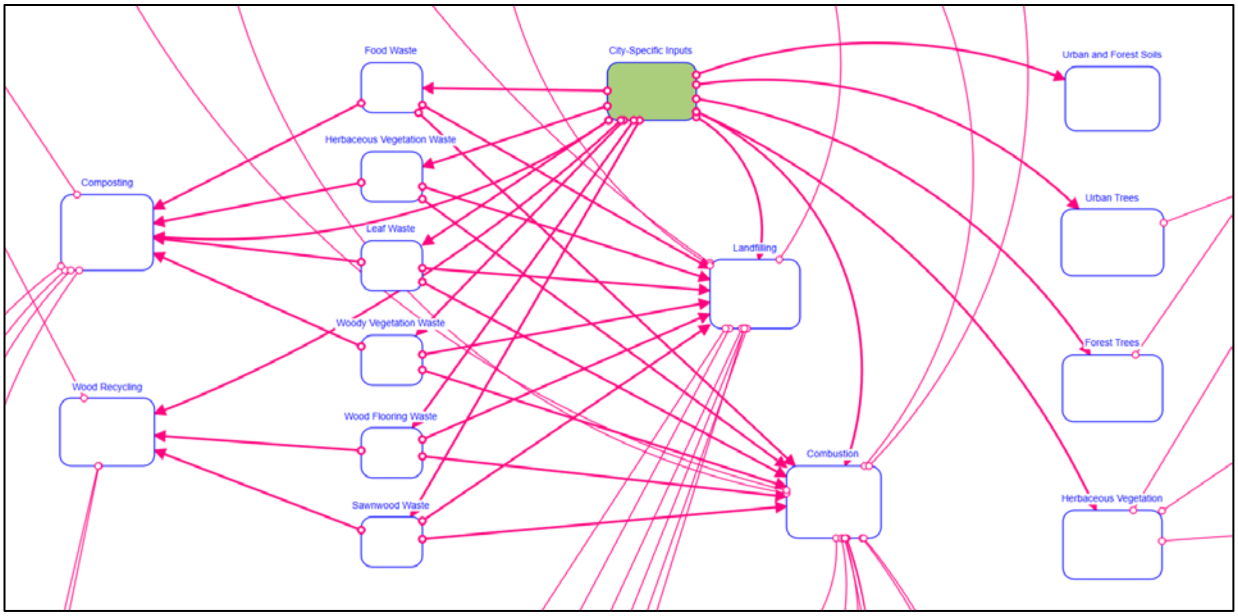

Figure 1: Screenshot of Model's Main Modules Representing Direct and Embodied Sinks

The model calculates carbon sequestration in direct sinks through modules for trees growing in forests within city boundaries, trees located in developed areas, herbaceous vegetation, and soil. The model allows a user to input predicted changes to the area of the city covered by tree canopy and herbaceous vegetation from year to year, such as from policies intended to expand tree cover and vegetation, or planned development projects expected to reduce tree cover and vegetation. Each module calculates the net impact of such changes on carbon sequestration over time.

The model calculates carbon sequestration in embodied carbon sinks through modules representing the waste management options for embodied carbon waste products, such as food waste, herbaceous yard waste, leaf yard waste, woody yard waste (i.e., deadwood from city forests and trees), and lumber and wood flooring waste from construction and demolition activities. The waste management modules include landfilling, composting, wood waste recycling, and burning waste products to generate energy. The model allows users to assign the percent of each waste product that is managed via each waste management practice. The modules calculate the net impact on carbon sequestration of such changes in a city's waste management practices over time.

Finally, the model calculates the total carbon sequestration across all direct and embodied carbon sinks. Some of the options for managing embodied carbon waste products involve using the products in place of fossil fuels, which does not remove carbon from the atmosphere and sequester it, but instead creates a carbon emissions offset. For simplicity, this study refers to both outcomes as "carbon sinks". The overall impact on a city's carbon balance from managing embodied carbon in such a way as to offset emissions changes and may even reverse as a city converts to using renewable energy sources, however. That is, when composting or burning a particular waste product reduces a city's carbon emissions by replacing fossil fuel use, the composting or burning process may become a net source of carbon emissions at such time as the city's energy supply is no longer generated from burning fossil fuels. The model accounts for this by allowing users to specify the emissions associated with the city's energy sources. Users can predict that these emissions will decrease over time, even to a 100% renewable energy future. The model results for modules representing embodied carbon sinks show how the effectiveness of these sinks may change over time, or even convert to carbon sources. The user can then change the model inputs to reflect possible future changes in waste management practices to maximize a city's carbon sequestration during a transition to renewable energy. This use of the model is described in more detail below.

The following sections detail the carbon sequestration calculations performed within each module for direct or embodied carbon sinks.

Direct Sinks

The IPCC recognizes three types of direct carbon sinks in urban areas: above- and below-ground biomass, dead organic matter (including both leaf litter and standing or fallen deadwood), and soil organic matter (Intergovernmental Panel on Climate Change (IPCC), Aalde, et al., 2006). Above- and below-ground biomass both store carbon during the life of the vegetation; however, this storage can be considered indefinite under the assumption that when individual plants die, they are replaced by a natural succession of similar vegetation. Dead organic matter releases its carbon back to the atmosphere over time and is therefore not a direct carbon sink for the purposes of this model (though dead organic matter may contribute to embodied carbon sinks if it is collected and added to urban waste streams, as discussed below). Soil organic carbon is considered to be a long-term direct carbon sink, with the degree of carbon sequestration dependent on the type of landcover, either trees or herbaceous vegetation (i.e., residential gardens, commercial and industrial green space, parkland, or vacant land consisting of grassland and/or shrubland).

The sections below discuss how the model calculates carbon sequestration in the above- and below-ground biomass of urban trees, the above- and below-ground biomass of forest trees within city limits, herbaceous vegetation, and urban and forest soils.

Urban trees.

The model calculates carbon sequestration by urban trees and forest trees separately, following IPCC guidance. IPCC Tier 2b methodology is used for estimating carbon sequestration by urban trees, defined in this study as trees growing in areas of the city zoned as residential, commercial, industrial, or public land. Since it is difficult to account for how recently a piece of vacant land was part of a formal settlement defined under the IPCC guidance, the model assumes that half of the tree cover on vacant land within city limits has the characteristics of urban trees, with the remaining vacant land treated as forested. Annual carbon sequestration by urban trees ($\Delta C_{urban\ trees}$) is calculated by multiplying the number of trees in each IPCC-defined species class ($N_{tree\ class}$) by the annual carbon sequestration rate for trees in that class ($C_{tree\ class}$), then summing the resulting carbon sequestration values for all tree species classes (Intergovernmental Panel on Climate Change (IPCC), Jenkins, et al., 2006). This calculation is shown below.

$$\boxed{\Delta C_{urban\ trees} = \sum_{tree\ class} N_{tree\ class} \cdot C_{tree\ class}}$$

The carbon sequestration rate for each tree species is provided by the IPCC and is summarized in Table 1 below. Values are given in megagrams of equivalent carbon dioxide ($MgCO_2eq$).

Table 1: Carbon Sequestration Rates for Urban Trees

Tree Type	**$C_{tree\ class}$ ($MgCO_2eq$)**
Hardwood trees	0.0100
Soft maple	0.0118
Hardwood maple	0.0142
Aspen	0.0096
Juniper	0.0033
Cedar/larch	0.0072
Douglas fir	0.0122
True fir/hemlock	0.0104
Pine	0.0087
Spruce	0.0092

The IPCC suggests calculating carbon sequestration only for trees that are less than 20 years old, on the assumption that after this point, a typical urban tree's rate of growth is equivalent to its losses to pruning and deadwood. However, the IPCC also advises adjusting this assumption to local circumstances. The model developed for this study assumes that urban trees sequester carbon only for 20 years, but the ability to designate some portion of urban trees as sequestering carbon for a longer period could be an addition to future iterations of the model.

For all urban trees, the model assumes that carbon losses due to deadwood and litter are at equilibrium. This is in keeping with IPCC methodology for urban lands that are not undergoing significant land use alterations (Intergovernmental Panel on Climate Change (IPCC), Jenkins, et al., 2006). As a result, the model does not calculate any lost carbon sequestration from deadwood or litter.

In calculating carbon sequestration by urban trees added to the landscape, the model takes as input the percent tree canopy expansion on impervious surfaces, vegetated surfaces (e.g., vacant lots), parkland, and residential land. These percentages are converted to specific numbers of trees by multiplying by the area of land in the appropriate category and by the average tree planting density in forested areas of the city. While tree planting density likely varies between urban trees and forest trees, data do not appear to be widely available to describe this density difference. This is a potential source of error in the model results and is discussed further below.

Forest trees.

Forested areas sequester carbon in trees, herbaceous plants, and soil. The model uses the gain-loss method within the IPCC Tier 1 methodology for estimating carbon sequestration by forest biomass (Intergovernmental Panel on Climate Change (IPCC), 2006). This method requires estimating the amount of carbon sequestered in the forest due to annual biomass growth in existing and newly planted forests, then subtracting the carbon lost from forests due to activities that remove live trees and/or deadwood. The IPCC distinguishes between intensively and extensively managed forests; this study assumes all forests with a city's jurisdiction are extensively managed. The model built for this study is aimed at suburban cities, which are unlikely to include intensively managed forests within their borders. The Fairfax County GHG inventory confirms that both agriculture and forestry are "virtually nonexistent" in the case study area (Bulova et al., 2013).

When applying this method to newly forested land, the model follows IPCC methodology by also accounting for the difference between the carbon sequestration of any vegetation removed when the forest was planted ($C_{cleared\ herbs}$), and the carbon sequestration of the vegetation in the new forest ($C_{forest\ herbs}$). These amounts are calculated using the method described in the herbaceous vegetation section below. This calculation is applied for 20 years after the conversion to forest, after which point it is assumed that forest understory vegetation has replaced the previously cleared herbaceous vegetation. All model calculations pertaining to carbon sequestration by newly planeted forest trees use IPCC estimates of biomass growth for "natural" forests, rather than "plantation" forests (Intergovernmental Panel on Climate Change (IPCC), 2006), based on the assumption that forests within city limits are typically too small to justify being managed as plantations.

Carbon sequestration due to annual growth of biomass is calculated in the model by multiplying the area of forested land (A_{forest}) by the total biomass growth in forests and the total fraction of carbon contained in local forest species. The area of newly forested land due to land cover changes is added to the total area on an annual basis. The IPCC provides tables that estimate the annual above-ground biomass growth of trees by species ($G_{above\text{-}ground}$), as well as the ratio of below-ground to above-ground biomass growth ($R_{below\text{-}to\text{-}above}$). The IPCC also provides tables that estimate the fraction of dry forest matter, by species, that is composed of carbon ($C_{tree\ wood}$) (Intergovernmental Panel on Climate Change (IPCC), 2006).

The IPCC provides different values for ($R_{below\text{-}to\text{-}above}$) based on the age of trees and the "tree class", i.e., whether they are oak species, other broadleaf hardwoods, or conifers. The model calculates carbon sequestration accordingly, using one ($R_{below\text{-}to\text{-}above}$) value for trees planted within the preceding 5 years, one ($R_{below\text{-}to\text{-}above}$) value for trees between 5-10 years old, and a final ($R_{below\text{-}to\text{-}above}$) value for established forest trees older than 10 years. In this way the model, like the IPCC methodology it utilizes, accounts for the U.S. EPA finding that hardwoods sequester more carbon than conifers (U.S. Environmental Protection Agency, Energy Information Administration, 1998), and also accounts for research by Nowak and Crane (2002) that found that carbon sequestration levels change as a forest ages.

The overall equation for calculating C_{forest} is shown below. IPCC values for the case study area are shown in Table 2.

$$\Delta C_{forest} = A_{forest} \cdot \left\{ \sum_{tree\ class} \left[G_{above-ground} \cdot (1 + R_{below-to-above}) \right] \right\} \cdot C_{tree\ wood} + A_{forest} \cdot \left(C_{forest\ herbs} - C_{cleared\ herbs} \right)$$

The equation above assumes that established trees more than 10 years old sequester carbon at the same rate. While some studies have found that younger trees sequester carbon at a more rapid rate than older trees (Pugh et al., 2019), other research has found that trees sequester more carbon as they grow larger (Stephenson et al., 2014). Due to the lack of scientific consensus on how carbon sequestration rates change as forest trees age, the model does not account for the possibility of such changes in carbon sequestration among established trees.

Table 2: Values Characterizing Carbon Sequestration by Forest Trees in Case Study Area

Variable	Value
$G_{above\text{-}ground}$	0.15 Mg/km^2
$R_{below\text{-}to\text{-}above}$: Conifers	0.20
$R_{below\text{-}to\text{-}above}$: Broadleaf	0.24
$R_{below\text{-}to\text{-}above}$: Quercus spp.	0.30
$C_{tree\ wood}$	0.48 $MgCO_2eq/Mg$
$D_{tree\ wood}$: Quercus spp.	0.58 Mg/m^3
$D_{tree\ wood}$: Acer spp.	0.52 Mg/m^3

$D_{tree\ wood}$: Other broadleaf	0.53 Mg/m^3

In terms of carbon losses from city forests, the model assumes that these arise only from land use change and from removing deadwood as a forest management activity. Logging or harvesting forest wood for fuel are not considered in the model on the assumption that city forests are typically not large enough to support these activities. This is certainly true for the case study area. Similarly, the model also does not include a method for estimating forest carbon losses due to forest fires or widespread insect damage. These factors would need to be added to the model in order to apply it to other geographic regions in which these factors are expected to have a significant impact.

The model accounts for the loss of carbon sequestration capability due to a reduction in the size of a city's forests by simply reducing the area of forested land that is used as a basis for calculating annual carbon sequestration in city forests. The model accounts for the loss of carbon sequestration capability from deadwood removal ($C_{deadwood\ removal}$) by multiplying the amount of deadwood removed ($G_{deadwood}$) by the estimated density of wood ($D_{tree\ wood}$) in local forests, then multiplying this result by the fraction of carbon in local wood ($C_{tree\ wood}$). The latter two values come from IPCC documentation (Intergovernmental Panel on Climate Change (IPCC), 2006); IPCC values for the case study area are shown in Table 2. The density of other broadleaf trees is not provided by the IPCC so this model uses an average of the IPCC values for hardwood species. The overall equation for calculating carbon losses from city forests is shown below.

$$C_{deadwood\ removal} = G_{deadwood} \cdot D_{tree\ wood} \cdot C_{tree\ wood}$$

Finally, the model does not account for the change in carbon emissions that results from discontinuing the use of fossil fuel-powered gardening equipment on land converted from parkland to forest, or initiating use of such equipment on land converted to parkland. Data are not available to enable an estimate of the carbon emissions associated with equipment used to maintain parkland.

Herbaceous vegetation.

The IPCC notes that data on the carbon sequestration provided by herbaceous vegetation are sparse but may be locally available (Intergovernmental Panel on Climate Change (IPCC), Jenkins, et al., 2006). The U.S. Federal Highway Administration (FHA) uses a value of 534 $MgCO_2eq/km^2/year$ for both grasses and shrubs (U.S. Department of Transportation, Federal Highway Administration, Office of Planning, Environment and Realty, 2010). FHA notes that this value is an average of a range provided by the former Chicago Climate Exchange. The model in this study uses the FHA value for all herbaceous vegetation.

Overall carbon sequestration by herbaceous vegetation ($C_{herbaceous\ veg}$) is calculated by multiplying the area of land with herbaceous vegetation cover ($A_{herbaceous\ veg}$) by the annual rate of carbon sequestration provided by such vegetation ($C_{grass/shrub}$). This equation is shown below.

$$C_{herbaceous\ veg} = A_{herbaceous\ veg} \cdot C_{grass/shrub}$$

Soil organic carbon.

A soil carbon sink is formed when land cover in a city converts from a type that sequesters comparatively less carbon to a type that sequesters comparatively more. The IPCC estimates that these changes in soil organic carbon levels reach an equilibrium state over a period of 20 years after the land cover change occurs (Intergovernmental Panel on Climate Change (IPCC), Aalde, et al., 2006); correspondingly, the model executes the changes over a 20-year time period. Values for carbon sequestration in the soil based on land cover types come from Pouyat, Yesilonis, and Nowak (2006) and are shown in Table 3. That study notes that the high values for parkland and residential land reflect management activities that increase soil organic carbon levels.

The model uses the soil carbon values in Table 3 to calculate the carbon sequestration (or emission) that results from city planners' decisions to change land cover types within the city. The table includes values applicable to states in the Southeastern region, which is the region that includes the case study area, and for states in the adjacent Mid-Atlantic region. Values for both regions are shown because the case study area is on the border between these two regions. This study includes a sensitivity analysis assessing the potential impact of this possible difference in values.

Table 3: Carbon Sequestration Rates for Urban Soil

Land Cover Type	$C_{soil\ organic\ carbon}$ ($MgCO_2eq/m^2$)	
	Southeastern Region	**Mid-Atlantic Region**
Impervious surface	$3.3 \cdot 10^{-9}$	$3.3 \cdot 10^{-9}$
Parkland	$7.1 \cdot 10^{-9}$	$7.1 \cdot 10^{-9}$
Residential land	$14.4 \cdot 10^{-9}$	$14.4 \cdot 10^{-9}$
Grassland/shrubland	$3.9 \cdot 10^{-9}$	$6.2 \cdot 10^{-9}$
Forest (remnant)	$7.7 \cdot 10^{-9}$	$11.6 \cdot 10^{-9}$
Forest (reforestation)	$6.2 \cdot 10^{-9}$	$9.3 \cdot 10^{-9}$

This study considers the impacts on soil organic carbon of converting existing grassland/shrubland or impervious surfaces to forested land, or converting existing impervious surfaces to parkland. These are the types of land cover changes that would typically be driven by city planners' environmental goals. While residential land management activities sequester the most carbon in soil, the possibility of increasing carbon sequestration is unlikely to act as a sole or primary driver for establishing new residential neighborhoods, so the model does not currently account for expansions in residential land area.

Calculating the increase in soil organic carbon due to land cover change (C_{soil}) requires multiplying the area of the affected land ($A_{land\ cover\ change}$) by the change in soil organic carbon ($C_{soil\ organic\ carbon}$) expected to occur following the land cover change. This result is divided by the 20-year timeframe in which the change takes place in order to obtain the annual rate of soil organic carbon increase. The final equation for C_{soil} is shown below and follows IPCC Tier 1 methodology (Intergovernmental Panel on Climate Change (IPCC), Aalde, et al., 2006).

$$C_{soil} = (A_{land\ cover\ change} \cdot C_{soil\ organic\ carbon}) / 20$$

Embodied Sinks

Embodied sinks derive from waste products that contain large amounts of carbon, such as food waste, yard waste (including both herbaceous yard trimmings such as grass and brush, and woody waste such as large branches), and wood waste from construction and demolition activities (including wood flooring and lumber). The carbon embodied in these products is biogenic: it was largely removed from the atmosphere and will return to the atmosphere upon the products' decomposition, unless human intervention sequesters the carbon; however, there are two ways policymakers can leverage this embodied carbon to create a carbon sink (U.S. EPA, Office of Resource Conservation and Recovery & ICF International, 2019a). The first involves sequestering the embodied carbon in a landfill under anaerobic conditions that partially arrest decomposition, thereby preventing some of the carbon from returning to the atmosphere. Embodied carbon can also be used to reduce fossil carbon use if the carbon is converted to an energy source or fertilizer that replaces the use of fossil fuels. For the purposes of this study, both storing embodied carbon and burning it to offset fossil fuel use are referred to as carbon sequestration. The sections below describe the calculations used to estimate the carbon sequestration potential of the various options for managing food and yard waste, including landfilling, composting, recycling, and combustion. The calculations are based primarily on the methodology of the U.S. EPA's Waste Reduction Model (WARM).

Landfilling.

The landfilling portion of the model calculates the carbon sequestration or emissions associated with sending each of the six waste types to a landfill managed for anaerobic decay. Storing embodied carbon waste products in an anaerobic landfill can partially arrest their decay, sequestering some of the embodied carbon indefinitely and thus creating a carbon sink.

The total carbon sequestered in a landfill ($C_{landfill\ sequestration}$) consists of the carbon expected to remain undecomposed in the landfill ($C_{undecomposed}$), minus the carbon emitted as CH_4 due to the partial decomposition of the waste product ($C_{landfill\ CH4}$), minus the carbon emitted during the processes of transporting the waste products to the landfill and operating landfill equipment ($C_{landfill\ transport\ \&\ ops}$). The equation for this calculation is shown below.

$$C_{landfill\ sequestration} = C_{undecomposed} - C_{landfill\ CH4} - C_{landfill\ transport\ \&\ ops}$$

$C_{undecomposed}$ varies based on the carbon content and decay rate of each waste product. EPA calculates the amount of undecomposed carbon for each waste product, converted to an avoided CO_2 emissions equivalent, as shown in Table 4 (U.S. EPA, Office of Resource Conservation and Recovery and ICF International 2019d; 2019b).

Table 4: Summary of Landfill Emissions Factors

Waste Type	$C_{undecomposed}$ ($MgCO_2eq/Mg$)	$C_{landfill\ CH4}$ ($MgCO_2eq/Mg$)			$C_{transport\ \&\ ops}$ ($MgCO_2eq/Mg$)
		no CH_4 flaring	with CH_4 flaring	CH_4 electric generation	
Food Waste	0.0992	-1.79	-0.694	-0.573	-0.022
Herb. Yard Waste	0.154	-0.562	-0.276	-0.254	-0.022
Leaf Yard Waste	0.871	-0.650	-0.287	-0.243	-0.022
Woody Yard Waste	1.17	-1.43	-0.717	-0.485	-0.022
Lumber Waste	1.17	-0.165	-0.0661	-0.0551	-0.022
Wood Flooring Waste	1.15	-0.176	n/a	n/a	-0.022

The CO_2-equivalent amount of CH_4 released to the atmosphere also varies by waste type and is shown in Table 4. EPA provides estimates for this value that differ based on whether the landfill allows all of its emitted methane to escape to the atmosphere, captures and flares the methane, or burns it to generate electricity (U.S. EPA, Office of Resource Conservation and Recovery and ICF International 2019d; 2019b). (Methane that is captured and flared or burned for electricity is oxidized to CO_2, which removes it from consideration in the model (U.S. EPA, Office of Resource Conservation and Recovery & ICF International, 2019a). Because this CO_2 is part of an ongoing cycle of carbon moving from plants to the atmosphere and back again, its emissions rate is not included in the model.) Since jurisdictions may send waste to multiple landfills which may differ in their handling of methane, the model allows for specifying the percent of waste that goes to landfills with each type of methane management. The one exception to this is wood flooring waste, which is generally managed only in construction and demolition-specific landfills that typically do not have any methane capture systems in place (U.S. EPA, Office of Resource Conservation and Recovery & ICF International, 2019b).

Finally, transportation emissions and landfill operation emissions also come from the EPA and are reported as combined values shown in Table 3 (U.S. EPA, Office of Resource Conservation and Recovery and ICF International 2019d; 2019b). In generating these values, EPA assumed the use of diesel trucks and equipment for waste management, and a waste transportation distance of 20 miles. The landfills in the case study region fall within this radius.

Composting.

Composting sequesters carbon when the compost is applied to land. This potentially leads to carbon sequestration in the soil but also reduces the amount of synthetic fertilizer used. This eliminates the CO_2 emissions from the fossil fuels typically used to manufacture synthetic fertilizer (Brown, 2016). Notably, this latter offset applies only while fossil fuels are used in the synthetic fertilizer lifecycle; if those fuels are replaced with renewable sources, this aspect of the carbon offset from using compost goes to zero.

The EPA reports that there is a lack of scientific consensus regarding the degree to which compost applications to land increase soil carbon. Using the Century model, EPA calculated that increases in soil carbon from compost applications are short-lived regardless of soil characteristics (U.S. EPA, Office of Resource Conservation and Recovery and ICF International 2019b). EPA notes that other researchers suggest the application of compost may change the entire process by which carbon cycles in and out of soil in a manner that increases soil organic carbon; however, EPA was unable to quantify this impact (U.S. EPA, Office of Resource Conservation and Recovery and ICF International 2019b). The increase in soil carbon that EPA includes in the WARM model is the increased soil carbon remaining 10 years after compost application to degraded agricultural soil, but EPA noted that this amount represented "only a fraction of the initial carbon added" (U.S. EPA, Office of Resource Conservation and Recovery and ICF International 2019c, page 4-5).

Using a different model, Hansen et al. (2006) calculated that the amount of carbon from compost that remained bound in the soil after 100 years (which the authors viewed as being bound in the soil indefinitely) was between 0.0 for loamy soils and 0.14 for sandy soils, as a fraction of the initial carbon present in the compost. The soils in the case study area are primarily clay or a mix of clay and loam. Based on the Hansen and EPA results, the model in this study assumes that no carbon is bound indefinitely in the soil due to application of compost. Boldrin et al. (2009) found that the greatest emissions reductions from compost arose when the compost was substituted for peat in growth mediums for gardening. As the estimated emissions reduction spanned a wide range of values, the model developed in the current study does not account for this possibility at this time.

The current study does apply a non-zero factor to estimate the carbon emissions offset created when compost displaces the use of synthetic fertilizers. To calculate this value, each Mg of waste is multiplied by a conversion factor that represents the megagrams of compost ($M_{compost}$) produced from composting the waste product. The conversion factors come from research done by the California Environmental Protection Agency and are summarized in Table 5 (California Environmental Protection Agency, 2017). The amount of natural fertilizer produced is then multiplied by a compost-to-fertilizer replacement rate ($R_{compost\text{-}to\text{-}fertilizer}$) estimated by Hansen et al. (2006), and finally by the estimated carbon-equivalent emissions rate per Mg of synthetic fertilizer produced ($C_{synthetic\ fertilizer}$) (Brown, Beecher, & Carpenter, 2010). This provides a value for the carbon-equivalent emissions avoided by replacing synthetic fertilizer with natural fertilizer ($C_{composting\ avoided\ emissions}$), as shown in the equation below.

$$C_{composting\ avoided\ emissions} = M_{compost} \cdot R_{compost\text{-}to\text{-}fertilizer} \cdot C_{synthetic\ fertilizer}$$

Table 5: Summary of Composting Emissions Factors

Waste Type	$M_{compost}$ (Mg)	$R_{compost\text{-}to\text{-}fertilizer}$	$C_{synthetic\ fertilizer}$ ($MgCO_2eq/Mg$)
Food Waste	0.55	0.30	-0.05
Herbaceous Yard Waste	0.66	0.30	-0.05
Leaf Yard Waste	0.66	0.30	-0.05
Woody Yard Waste	0.66	0.30	-0.05

Emissions from the composting process include the CH_4 and N_2O generated by the composting materials, CO_2 emitted by the trucks that transport waste and compost to/from the compost facility, and CO_2 emitted by the fuel for the equipment needed to perform composting operations (U.S. EPA, Office of Resource Conservation and Recovery and ICF International

2019b). Komilis and Ham (2004) found that the vast majority of CO_2 emissions from composting operations arose from the decomposition process, while the remaining less than 10% came from the use of fossil fuels to power composting equipment. The current study models emissions from the decomposition process ($C_{composting\ decomposition\ emissions}$) by using estimates developed for the EPA WARM model, which are 0.0551 $MgCO_2eq/Mg$ food waste and 0.0772 $MgCO_2eq/Mg$ yard waste (including leaves, woody yard waste, and herbaceous yard waste) (U.S. EPA, Office of Resource Conservation and Recovery and ICF International 2019b).

The emission factor associated with transporting waste to/from the composting site ($C_{composting\ transportation\ emissions}$) also comes from the WARM model and is 0.0034 $MgCO_2eq/Mg$ (U.S. EPA, Office of Resource Conservation and Recovery and ICF International 2019). This estimate assumes that the waste is transported using a diesel waste collection truck over a distance of 20 miles. The emissions would go to zero if/when a city adopted a clean fuel source for such trucks.

The emissions factor associated with composting operations ($C_{composting\text{-}ops}$) is dependent on the composting process and fuel required to operate composting equipment. The model assumes open-air composting in windrows. This is the method used in the case study area, where the local private compost company (Compost Crew) transports food waste from the region to the Prince George’s County composting facility in Maryland (Maryland Environmental Service, Gibson, Rybak, Curry, & Birchfield, n.d.; Prince George’s County, MD, n.d.). The emissions from operating this facility arise from burning fossil fuels for electricity and as fuel for equipment, such as the machines that turn the compost piles. Komilis and Ham (2004) calculate the energy required for windrow composting ($E_{composting\ ops}$) at 167 kWh/Mg for composting food waste and 29 kWh/Mg for composting yard waste. The EPA provides regional estimates of carbon emissions reductions from fossil fuel use (U.S. EPA, Office of Resource Conservation and Recovery & ICF International, 2019c). The full carbon-equivalent emissions ($C_{composting\ ops}$) from composting each Mg of food and yard waste ($M_{food/yard\ waste}$) is calculated as follows:

$$C_{composting\text{-}ops} = M_{food/yard\ waste} \cdot E_{composting\ ops} \cdot C_{fossil\ fuel\ combustion}$$

The total carbon sink (or source) created by composting waste is then calculated by subtracting emissions from waste decomposition, waste transport to the composting facility, and composting facility operations, from the emissions avoided by replacing synthetic fertilizer with natural fertilizer. This calculation is shown in the equation below.

$$C_{composting} = C_{composting\ avoided\ emissions} - C_{composting\ decomposition\ emissions} - C_{composting\ transportation\ emissions} - C_{composting\ ops}$$

Recycling.

The net carbon-equivalent sequestration that results from recycling wood waste ($C_{recycling}$) is calculated as the difference between emissions from harvesting and transporting new wood ($C_{virgin\ wood\ harvesting}$ and $C_{virgin\ wood\ transport}$), and the emissions from the process of recycling and transporting wood waste ($C_{wood\ waste\ processing}$ and $C_{wood\ waste\ transport}$). The carbon sequestered by the recycled wood at its end-of-life date ($C_{retired\ wood\ waste}$) is also added to this total. The model assumes that wood products are recycled only once, and that they have a 50-year lifespan as recycled products. At the end of that time period, the model assumes the products are burned in a WTE facility or landfilled at a ratio that matches that of current waste management practices at the time. Any fraction of retired recycled wood that matches that recycling rate of the time period is assumed to be landfilled. The final calculation is shown in the equation below:

$$C_{recycling} = (C_{wood\ waste\ processing} + C_{wood\ waste\ transport}) - (C_{virgin\ wood\ harvesting} + C_{virgin\ wood\ transport}) + C_{retired\ wood\ waste}$$

The values for these variables in the model result in negative carbon sequestration during the lifetime of the recycled product, indicating that recycling wood waste results in net carbon emissions as compared to obtaining wood by harvesting trees. EPA (2019a) calculates that transporting recycled wood in diesel trucks for an average distance of 20 miles results in higher emissions than transporting newly harvested wood, by a factor of 0.011 $MgCO_2eq/Mg$ wood waste. Similarly, EPA found that the process of recycling wood waste emitted more carbon than the process of harvesting new wood, by a factor of 0.0661 $MgCO_2eq/Mg$ wood waste.

Other researchers (Morris, 2017; U.S. EPA, Office of Resource Conservation and Recovery & ICF International, 2019b) have found that recycling wood waste results in net carbon sequestration when the recycling increases carbon uptake by forests that would have been harvested in the absence of a wood waste recycling program. The current model assumes that any harvested wood would come from managed forests where trees are replaced by new plantings after harvest, thereby keeping annual carbon uptake in forests substantially unchanged regardless of the number of trees harvested. Further, the carbon stored in existing forests is released when the trees are harvested or otherwise die, so the forest does not act as a carbon sink unless it is increasing in tree density. EPA notes that forest managers only plant new trees to replace those removed from the forest, but do not increase tree density in a managed forest when demand for trees decreases (U.S. EPA, Office of Resource Conservation and Recovery & ICF International, 2019a). Therefore, to the extent that recycling wood products reduces the demand for wood from managed forests, it does not affect carbon uptake in those forests.

Finally, the model does not provide any carbon sequestration credit for the carbon embodied in the products made from recycled wood. This is because the model focuses on the final destination of the carbon embodied in waste products, and products made from recycled wood have not yet reached their end of life.

Combustion.

There are two methods by which a city can burn waste to generate energy: using waste-to-energy (WTE) facilities or using industrial boilers (Morris, 2017). In each case, the energy generated from combustion is assumed to replace electricity. This represents avoided carbon-equivalent emissions to the extent that the electricity generated from waste reduces the amount of electricity that must be generated by burning fossil fuels. As a jurisdiction draws more of its energy from renewable sources and less from fossil fuels, the potential carbon emissions offset from burning waste to produce energy decreases. The model accounts for this by allowing the variable representing the amount of fossil fuels the jurisdiction burns for energy to decrease over time, and by giving the jurisdiction a carbon credit that is no greater than the total amount of potential carbon emissions from fossil fuels in any given year.

Waste-to-Energy Combustion.

Burning waste to generate electricity results in carbon-equivalent emissions ($C_{wte\ combustion}$) from transporting the waste to the WTE facility ($C_{wte\ transport}$), burning fossil fuels to operate equipment within the facility ($C_{wte\ ops}$), and emitting N_2O generated during the combustion process ($C_{wte\ N2O\ emissions}$). The process also sequesters carbon by replacing fossil carbon (including coal and natural gas) as a fuel source for generating electricity ($C_{wte\ avoided\ emissions}$). This calculation is shown in the equation below.

$$\boxed{C_{wte\ combustion} = C_{wte\ avoided\ emissions} - C_{wte\ transport} - C_{wte\ ops} - C_{wte\ N2O\ emissions}}$$

Emissions from operating a WTE facility are estimated by Morris (2017) as 0.032 $MgCO_2eq/Mg$ wood waste. This study uses this same value for all types of waste, on the assumption that the same equipment is used for processing all types of waste. EPA provides values for transportation and N_2O emissions for each waste product included in this study, as shown in Table 6 (U.S. EPA, Office of Resource Conservation and Recovery and ICF International 2019d; 2019b). The transportation emissions factors assume the use of short-haul diesel trucks to transport waste to the combustion facility and ash to the landfill, over distances of 20 miles (U.S. EPA, Office of Resource Conservation and Recovery and ICF International 2019c). N_2O emissions values reflect emissions from mass burn facilities, which comprise over three-quarters of the WTE facilities in the U.S. (U.S. EPA, Office of Resource Conservation and Recovery and ICF International 2019c), including the WTE facilities serving the case study area (Michaels, n.d.).

Table 6: Summary of Combustion Emissions Factors

Waste Type	**$C_{combustion\ N2O\ emissions}$ ($MgCO_2eq/Mg$)**	**$C_{transport}$ ($MgCO_2eq/Mg$)**	**$R_{wte\ conversion\ to\ electricity}$**	**$E_{wte\ energy\ content}$ (million Btu/Mg)**
Food Waste	-0.0441	-0.0110	0.178	5.18
Herbaceous Yard Waste	-0.0441	-0.0110	0.178	6.17
Leaf Yard Waste	-0.0441	-0.0110	0.178	6.17
Woody Yard Waste	-0.0441	-0.0110	0.178	6.17
Lumber Waste	-0.0441	-0.0110	0.178	18.3
Wood Flooring Waste	-0.0441	-0.0551	0.215	19.8

The net carbon-equivalent emissions avoided by replacing fossil fuels with WTE in electricity production ($C_{wte\ avoided\ emissions}$) is calculated by multiplying the amount of electricity generated from WTE plant operations ($E_{wte\ combustion}$) by the emissions that would be generated by burning the same amount of fossil fuels ($C_{fossil\ fuel\ combustion}$). $E_{wte\ combustion}$ is calculated for each waste type by multiplying the amount of waste combusted ($M_{combustion}$) by the energy content of that waste type ($E_{wte\ energy\ content}$) and its combustion efficiency ($R_{combustion\ efficiency}$). These calculations are shown below.

$$C_{wte\ avoided\ emissions} = E_{wte\ combustion} \cdot C_{fossil\ fuel\ combustion}$$

$$\text{where } E_{wte\ combustion} = M_{combustion} \cdot E_{wte\ energy\ content} \cdot R_{combustion\ efficiency}$$

EPA provides values for $E_{wte\ energy\ content}$ and $R_{combustion\ efficiency}$ for each waste type, as shown in Table 6 (U.S. EPA, Office of Resource Conservation and Recovery and ICF International 2019d; 2019b). EPA also provides a rate of efficiency for generating energy from natural gas ($R_{natural\ gas\ combustion}$) of 264.5 Btu/m^3 and a rate of efficiency for generating energy from coal ($R_{coal\ combustion}$) of 5.9 Btu/Mg (U.S. Environmental Protection Agency, n.d.). For this study, bituminous coal use was assumed as this is the most widely combusted type of coal in the U.S. (U.S. Geological Survey, n.d.).

Industrial Boiler Combustion.

Burning wood waste in the form of wood chips in an industrial boiler to generate electricity results in emissions ($C_{boiler\ combustion}$) from transporting the waste to the boiler ($C_{boiler\ transport}$), and from the combustion process ($C_{boiler\ non\text{-}CO2\ emissions}$). Combustion in an industrial boiler also sequesters carbon by replacing fossil carbon (including coal and natural gas) as a fuel source for generating electricity ($C_{boiler\ avoided\ emissions}$). This calculation is shown in the equation below. (Emissions from operating an industrial boiler are not included in the calculation because they are assumed to be the same regardless of the type of boiler fuel used, such that replacing fossil boiler fuels with wood chips does not change emissions.)

$$C_{boiler\ combustion} = C_{boiler\ avoided\ emissions} - C_{boiler\ transport} - C_{boiler\ non\text{-}CO2\ emissions}$$

EPA calculates that transportation of wood waste to an industrial boiler results in emissions of 0.010 $MgCO_2eq$/Mg wood waste (U.S. EPA, Office of Resource Conservation and Recovery & ICF International, 2019b). The transportation emissions factor assumes the use of short-haul diesel trucks to transport waste to the combustion facility and ash to landfills (U.S.

EPA, Office of Resource Conservation and Recovery and ICF International 2019c). Emissions from burning wood waste in an industrial boiler are estimated by Morris (2017) as 0.032 $MgCO_2eq/Mg$ waste.

The net carbon-equivalent emissions avoided by replacing fossil fuels with wood chips as fuel for industrial boilers ($C_{boiler\ avoided\ emissions}$) is calculated by multiplying the amount of industrial boiler wood chip fuel generated from wood waste ($M_{wood\ chips}$) by the rate of efficiency at which wood chips replace each fossil fuel type ($R_{wood\ replacing\ natural\ gas}$ and $R_{wood\ replacing\ coal}$) and by the emissions rate from burning that fossil fuel ($C_{natural\ gas\ boiler\ combustion}$ and $C_{coal\ boiler\ combustion}$). $M_{wood\ chips}$ is calculated by multiplying the amount of each type of wood waste ($M_{wood\ waste}$) by the percent of that waste converted to wood chips for burning in a boiler ($P_{wood\ waste\ to\ boiler}$). These equations are shown below.

$$C_{boiler\ avoided\ emissions} = M_{wood\ chips} \cdot (R_{wood\ replacing\ natural\ gas} \cdot C_{natural\ gas\ boiler\ combustion} + R_{wood\ replacing\ coal} \cdot C_{coal\ boiler\ combustion})$$

$$\text{where } M_{wood\ chips} = M_{wood\ waste} \cdot P_{wood\ waste\ to\ boiler}$$

Efficiencies for converting wood waste to wood chips for industrial boiler fuel come from Morris (2017) and are 0.53 Mg coal / Mg wood chips and 379 m^3 natural gas / Mg wood chips. Morris also provides values for the carbon-equivalent emissions of burning natural gas and coal in industrial boilers, which amount to 3.033 $MgCO_2eq/Mg$ coal and 0.00229 $MgCO_2eq/m^3$ natural gas.

Case Study: Fairfax County

The model was applied to Fairfax County, Virginia, primarily using data available from County government reports. The tables below summarize the input values used, and their sources, for each direct and embodied carbon sink module.

Direct Carbon Sinks.

Data describing land cover and vegetation characteristics came from several Fairfax County government reports. The 2016 Fairfax County Land Use and Transportation report provides the total acreage of County land as assigned to the following land use categories: residential, industrial, commercial, public facilities, parks and recreation (which includes forested land), and vacant land (Fairfax County Environmental Quality Advisory Council, 2016). The 2017 Fairfax County Land Cover Change Analysis report provides the percentages of residential, commercial, and industrial land that is either already tree canopy, or could be converted from either vegetated or impervious surface to tree canopy (O'Neill-Dunne, University of Vermont, Spatial Analysis Laboratory, & Fairfax County, 2017). Percentages for public facilities were not given, so this study used the percentages provided for commercial land. This approximation is based on the assumption that public facilities (which include government buildings, fire stations, police stations, landfills, etc.) are most similar to commercial facilities.

Percentages were also missing for parkland and vacant land. This study approximated the percentages as follows. First, the total acres of residential, commercial, industrial, and public land in each of the categories of existing tree canopy, possible tree canopy on vegetated land, and possible tree canopy on impervious land were calculated. These results were subtracted from the County-wide total of land falling within each of the three land cover categories. This provided the acres of land in each of the three categories that must be either parkland or vacant land. The unassigned acres in each category were then assigned to parkland or vacant land in such a way as to preserve the ratio of parkland to vacant land within each category.

Table 7 summarizes the area of land included in each land use category, as well as the percent of that land that is currently covered in tree canopy or could be converted to tree canopy, as discussed above. According to the 2017 Fairfax County Land Cover Change Analysis report (O'Neill-Dunne et al., 2017), vegetated land that is categorized as a possible site for future tree canopy includes land that is unlikely to be converted to trees, such as athletic fields. Similarly, although the category of impervious surfaces that could be converted to tree canopy does not include buildings or roads, presumably it does include parking lots and other areas that were paved for a purpose. This study therefore assumes that policies aimed at converting a high percentage of the land categorized as possible future tree canopy would face significant opposition in any land use category other than vacant land. As a result, tree planting scenarios considered in this study are modest in scope, as discussed below.

Table 7: Fairfax County Land Cover by Land Use Category

Land Cover	Residential	Commercial	Industrial	Public	Parkland	Vacant
Total Hectares of Land	53367.2	4694.8	4106.8	10292.4	13539.6	6216.4
Existing Tree Canopy	60%	21%	30%	30%	63%	63%
Possible Tree Canopy (vegetated land)	23%	11%	18%	18%	22%	22%
Possible Tree Canopy (impervious surface)	6%	42%	32%	32%	4%	4%
No Tree Canopy Possible	11%	26%	20%	20%	11%	11%

Calculations of carbon sequestration within urban and forest trees depend on local tree species distribution, which was provided for this study by the i-Tree Ecosystem Analysis report for Fairfax County (Fairfax County, 2018). The same report also provided an estimate of the total number of trees in the County. These data are shown in Table 8. The table also displays the IPCC categories to which each tree belongs for different steps of the model calculations.

One species of tree that is among the more common in the County, the Eastern Red Cedar (which provides 4.5% of the tree cover (Fairfax County, 2018)), did not fall into any of the categories for which the IPCC provides estimated wood density. This study assigned Eastern Red

Cedar trees a density corresponding to the average of the densities provided by the IPCC for eight other conifer species (Intergovernmental Panel on Climate Change (IPCC), 2006).

Table 8: Tree Species Distribution in Fairfax County

Tree Species	Tree Canopy Contribution	IPCC Carbon Sequestration Category Used (urban trees)	IPCC Ratio of Below-to-Above Ground Biomass Growth Category Used (forest trees)	IPCC Density of Tree Wood Category Used (forest trees)
American beech	9.7%	mixed hardwood	broadleaf trees	Fagus
Red maple	9.7%	soft maple	broadleaf trees	Acer
Tulip tree	6.3%	mixed hardwood	broadleaf trees	broadleaf trees
Black tupelo	6.2%	mixed hardwood	broadleaf trees	broadleaf trees
White oak	5.3%	mixed hardwood	Quercus	Quercus
Eastern red cedar	4.5%	juniper	conifers	conifer average
Green ash	3.8%	mixed hardwood	broadleaf trees	Fraxinus
American hornbeam	3.6%	mixed hardwood	broadleaf trees	broadleaf trees
Sweetgum	3.5%	mixed hardwood	broadleaf trees	broadleaf trees
American holly	3.2%	mixed hardwood	broadleaf trees	broadleaf trees
Other	44.1%	mixed hardwood	broadleaf trees	broadleaf trees

Embodied Carbon Sinks.

Data describing Fairfax County's yard waste, leaf waste, lumber waste, and wood flooring waste came primarily from the County's 2015 update to its Solid Waste Management Plan (SWMP). The 2015 SWMP updates states that herbaceous and leaf yard waste, combined, has historically comprised 9.4% of the County's total municipal solid waste (MSW) (Fairfax County, 2004a). The updated SWMP provides two projections for total MSW: one that assumes the per capita generation of waste stays constant at 2013 rates, and one that assumes the per capita rate increases over time. This study uses the latter projections based on the County's expectation that the 2013 waste generation rate was an outlier. The County provides MSW projections through 2035 at a linear rate of change; this study extrapolated values through 2100 using the same rate. Combined herbaceous and leaf yard waste projections were calculated based on the resulting total MSW projections. This study then estimated that approximately 90% of mixed yard waste is herbaceous and 10% consists of leaves; this ratio was based on averaging data from 2000-2002 (Fairfax County, 2004b). The same 2000-2002 data set showed that approximately 94% of County herbaceous yard waste and leaf waste was composted; this study uses this percentage for all years in the baseline case and assumes that the remaining 6% is burned in the County's waste-to-energy facility.

County data describing construction and demolition debris (CDD) provided the basis for estimating lumber waste and wood flooring waste. The original SWMP provided projected a CDD generation rate for 2020 and 2025 on a per person, per year basis (Fairfax County, 2004a). This study uses the 2025 rate for all years between 2025-2100; this rate is lower than the 2020 rate and therefore represents a more conservative view of construction in the County. This rate was multiplied by Fairfax County's projected population, which is given in 5-year increments from 2020-2045 and is assumed to continue growing at a linear rate thereafter (Fairfax County Department of Management and Budget, Economic, Demographic and Statistical Research, Han, Hovland, & Khaja, 2018). Fairfax County does not publish data regarding the estimated amount of lumber or wood flooring waste within its CDD waste stream, so the EPA's national-level estimates of 5.0% and 1.6%, respectively, were used in this study (U.S. EPA, Office of Resource Conservation and Recovery, 2016). The latter value is for wood paneling, which this study assumes would include wood flooring along with other, similar products. Fairfax County projects recycling rates for wood through 2035 using a linear rate of change, which this study used to extrapolate data through 2100 (Fairfax County, 2015). This study finds the average projected recycling rate over this time period, then multiplies this rate by the amount of wood projected to be in the County waste stream. The study assumes that all of this recycled wood is lumber, on the basis of an IPCC note that recycling wood flooring is extremely rare (U.S. EPA, Office of Resource Conservation and Recovery & ICF International, 2019b). Remaining lumber waste and all wood waste are assumed to be sent to the Fairfax County WTE facility.

Woody yard waste is assumed to consist of deadwood (both large tree branches and tree trunks) removed from County forests and parks. The Fairfax County office of the Virginia Department of Forestry states that any wood waste collected from County parks and forests is already included in MSW estimates for the County under yard waste (J. McGlone, personal communication, November 21, 2019). Without a method for estimating the amount of wood waste within this yard waste value, this study simply assumes that wood waste removed from County forests is zero.

Fairfax County does not measure food waste directly. This study uses a national estimate of food waste as a percentage of total municipal solid waste, as does the County in its own MSW projections (Fairfax County, 2004a). The percentage used is 15.2%, which is the average amount of food waste as a percent of total MSW at the national level from 2015-2017, as provided by the EPA (U.S. EPA, 2019). This was multiplied by the same MSW projections for Fairfax County that were used to estimate herbaceous yard waste and leaf waste. Since Fairfax County's composting program is in its infancy, this study assumed all food waste is sent to the County's WTE facility in the baseline case.

Table 9 summarizes both the amount of waste in each category (as a percentage of Fairfax County's total MSW) and the percent of each waste type that is currently disposed of through landfilling, composting, recycling, or burning.

Table 9: Fairfax County Waste Stream Data

Waste Type	% of MSW	% Landfilled	% Composted	% Recycled	% Burned
Herb. Yard Waste	8.46%	0%	94%	n/a	6%
Leaf Waste	0.94%	0%	94%	n/a	6%
Food Waste	15.2% (est.)	0%	0%	n/a	100%
Wood Yard Waste	n/a	0%	n/a	n/a	100%
Lumber	5.0% of CDD	0%	n/a	18.6%	81.4%
Wood Flooring	1.6% of CDD	0%	n/a	0%	100%

In addition to waste stream data, model calculations of avoided carbon emissions require data on Fairfax County's emissions due to fossil fuel use for electricity and composting. The EPA estimate for carbon reductions due to replacing fossil fuel use in South Atlantic states, which include Virginia, is 0.231 $MgCO_2eq/mmBtu$. This value also applies to Maryland, which is the site of composting facilities used by Fairfax County's nascent composting service provider, Compost Crew. Maryland has passed legislation that requires the state to obtain 50% of its electricity from renewable sources by 2030 (U.S. Energy Information Administration n.d.); in 2018, the state obtained 44% of its energy from such sources (including nuclear power plants) (U.S. Energy Information Administration n.d.). The model proportionately reduces the reduction in emissions from fossil fuel combustion from 2020-2030 to reflect this policy implementation.

Results

The model was run for Fairfax County in three scenarios:

- Baseline Scenario: carbon sequestration in Fairfax County based on existing direct sinks and embodied carbon waste management practices.
- Maximized Carbon Sequestration Scenario: potential maximum carbon sequestration from possible policies to increase the size of direct carbon sinks and optimizing waste disposal practices for embodied carbon products.

- Transition to Renewable Energy Scenario: impact on waste disposal practices for sequestering maximum carbon as the County transitions to renewable energy sources, and for transporting and processing embodied carbon waste products.

Baseline Scenario

The model estimates that direct carbon sinks in Fairfax County sequester 148,772 $MgCO_2eq$/year. Forests sequester 115,576 $MgCO_2eq$/year of this total through annual tree growth. Established urban trees do not contribute to annual carbon sequestration due to the assumption that annual tree growth is at equilibrium with deadwood removal, pruning, and trimming. Carbon stored in both forest and urban soils is also at equilibrium in the baseline case due to the lack of land cover change; therefore, the model finds no annual carbon sequestration in forest or urban soils. The model estimates that herbaceous urban vegetation in Fairfax County sequesters 33,195 $MgCO_2eq$/year. These finding are summarized in Table 10.

Table 10: Fairfax County Carbon Sequestration in Direct Sinks: Baseline Scenario

Direct Urban Carbon Sink	Annual Carbon Sequestration ($MgCO_2eq$)
Forest trees	115,576
Forest soil	0
Urban trees	0
Urban herbaceous vegetation	33,195
Urban soil	0
Total Carbon Sequestration in Direct Sinks	**148,772**

The model estimates that management of embodied carbon in Fairfax County prevents the release of 47,391 $MgCO_2eq$ in 2020. This amount increases over time as the population – and therefore the waste generated – increases. In the baseline scenario, the model calculates that the County's WTE facility produces electricity sufficient to prevent the release of 47,391 $MgCO_2eq$ in 2020. The model calculates that zero $MgCO_2eq$ is sequestered in landfills, since Fairfax

County sends its waste to WTE facilities. Both recycling and composting emit more GHGs than they sequester. The model estimates that composting emits 4,759 $MgCO_2eq$ in 2020, while recycling wood products emits 477 $MgCO_2eq$ in 2020. The values increase over time as the amount of waste generated increases. These findings are summarized in Table 11.

Table 11: Fairfax County Embodied Carbon Sequestration: Baseline Scenario

Embodied Carbon Source	**Carbon Emissions Offset ($MgCO_2eq$)**	
	2020	**2045**
Biofuels	52,627	69,162
Landfills	0	0
Composting	*-4,759*	*-7,583*
Wood Recycling	*-477*	*-546*
Total Carbon Emissions Offset	**47,391**	**61,506**

Maximized Carbon Sequestration Scenario

The model was used to generated estimates of the impact on carbon sequestration of changes in land cover and waste management practices for embodied carbon waste products. The model was run for three tree planting campaigns, described below and detailed in Table 12:

- Modest Tree Planting Campaign: in this scenario, the County increases tree cover primarily on parkland and vacant land, and to a much smaller extent on already vegetated residential, commercial, industrial, and public land.
- Aggressive Tree Planting Campaign: in this scenario, the County increases tree cover by twice as much on parkland and vacant lands, and by close to twice as much on vegetated residential, commercial, industrial, and public land. This scenario also models a small-scale campaign to replace impervious surfaces with trees in the latter four land use categories.
- Balanced Tree Planting Campaign: this scenario recognizes the limitations of tree planting campaigns, particularly on lands that are already under continuous use. It

therefore uses the modest tree planting campaign estimates for parkland and for vegetated land in all other land use categories except vacant lands. On vacant lands, the aggressive tree planting campaign is implemented. On commercial and industrial lands, a small amount of impervious surface is considered converted to tree cover.

Table 12: Tree Planting Campaign Scenarios

	Percent Increase in Tree Cover					
	Modest		**Aggressive**		**Balanced**	
Land Use	**vegetated**	**impervious**	**vegetated**	**impervious**	**vegetated**	**impervious**
Parkland	10%	5%	20%	10%	10%	5%
Vacant Land	25%	25%	50%	50%	50%	50%
Residential	2%	0%	5%	0%	2%	0%
Commercial	3%	0%	5%	1%	3%	1%
Industrial	3%	0%	5%	1%	3%	1%
Public	5%	1%	10%	5%	5%	1%

The results of the three model runs are shown in Table 13 in terms of both annual carbon sequestration, and in terms of the difference from the baseline scenario.

Table 13: Fairfax County Carbon Sequestration in Direct Sinks: Maximized and Balanced Scenarios

	Annual Carbon Sequestration ($MgCO_2eq$)	
Direct Urban Carbon Sink	**Total**	**Difference from Baseline**
Modest Tree Planting Campaign	161,528	+12,756
Aggressive Tree Planting Campaign	174,090	+25,318
Balanced Tree Planting Campaign	169,706	+20,934
Baseline Scenario	**148,772**	-

Model runs to identify the maximum carbon sequestration from managing embodied carbon in waste streams focused on calculating the results of managing each type of waste through landfilling, burning in a WTE facility, or composting or recycling as applicable. The landfilling option in these model runs assumed the landfill captures methane and flares it to generate energy. During the most recent shutdown of the Fairfax County WTE plant for repairs, in 2017, the County sent all of its waste to the King George Landfill (Fairfax County, n.d.), which captures methane gas and burns it for energy (Waste Management, Inc., 2009). For this study, the model runs therefore assumed that County waste would go to the King George Landfill in the absence of the WTE facility.

Table 14 shows the results of the model runs for the year 2020. Food waste is already managed for maximum carbon offsets by being burned in the WTE facility. The County could achieve greater carbon offsets by burning all of its herbaceous yard waste and leaf yard waste. Wood flooring and lumber waste sequester carbon most effectively in a landfill. If the County implemented all of these options together, the model predicts that it could sequester or offset emissions of 78,350 $MgCO_2eq$ per year. In the case that the County's practice of composting leaf yard waste is unlikely to change due to the considerations that initiated the composting program, this study considered a balanced case in which no change is made to the County's management of herbaceous or leaf yard waste, but all food waste is sent to the WTE facility and all wood waste is sent to a CDD landfill. This resulted in carbon sequestration of 76,130 $MgCO_2eq$ per year.

Table 14: Fairfax County Embodied Carbon Sequestration: Maximized and Balanced Scenarios

Waste Management Scenario	**2020 Carbon Emissions Offset ($MgCO_2eq$)**	
	Total	**Difference from Baseline**
Food Waste		
- Compost 90%; Burn 10%	42,147	-5,244
- Landfill 100%	-738,341	-785,732
Herbaceous Yard Waste		
- Burn 100%	**67,371**	+19,980
- Landfill 100%	-80,393	-127,784
Leaf Yard Waste		
- Burn 100%	**49,611**	+2,220
- Landfill 100%	39,600	-7,791
Wood Flooring Waste		
- Recycle 100%	37,480	-9,911
- Landfill 100%	**48,389**	+998

Lumber Waste		
- Burn 100%	51,983	+4,592
- Recycle 100%	27,295	-20,096
- Landfill 100%	**55,152**	+7,761
Baseline Scenario	**47,391**	-
Maximized Scenario	**78,350**	**+30,959**
Balanced Scenario	**76,130**	**+28,739**

The overall carbon sequestration in direct and embodied carbon sinks yielded the results for the County shown in Table 15. For illustrative purposes, these changes are assumed to take effect immediately, in 2020.

Table 15: Fairfax County Maximized and Balanced Carbon Sequestration

Scenario	**2020 Carbon Sequestration and Offsets ($MgCO_2eq$)**	
	Total	**Difference from Baseline**
Baseline Scenario	196,163	-
Maximized Scenario	252,440	+56,277
Balanced Scenario	245,836	+49,673

Transition to Renewable Energy Scenario

This study also explored optimal waste management practices for a scenario in which Fairfax County gradually converts to renewable energy sources and to zero emissions vehicles for waste transport. Under the Clean Power Plan of the Obama administration, Virginia was planning to reduce its power-related emissions by 32% by 2030 (Cleveland, Shepherd, & Beall, n.d.). The model simulates implementing this plan by reducing power-related emissions linearly from 2020-2030 to reach 32% of 2020 power-related emissions. Extending this scenario to a zero emissions future for the County, the model reduces power-related emissions by an additional 32% from 2030-2050, then to zero emissions by 2070.

Using these values, the model predicts that the carbon offset from Fairfax County's current waste management practices drops to zero midway through 2047. After this point, the

County's management of embodied carbon waste begins to result in net emissions. The model shows that the County can sequester some of this embodied carbon by altering its waste management practices. Results of model runs testing different waste management practices are shown in Table 16.

The set of options that maximizes carbon sequestration involves undertaking a tree planting campaign, implementing a food composting program, burning all herbaceous and leaf yard waste, and landfilling all wood waste. To simulate this, the model was run using the modest tree planting campaign defined previously, but enacted in 2047. It was also assumed that Fairfax County adopts food waste composting program in 2033 and achieves 90% composting of food waste by 2046. Sending all yard waste to the WTE facility and landfilling all wood waste are assumed to take place in 2020, as these changes can be implemented immediately. The model predicts that this combination of options would sequester 15,265,874 $MgCO_2eq$ by 2100, an increase of 4,508,084 $MgCO_2eq$ over the baseline scenario.

The model was also run as above but with current levels of composting for herbaceous yard waste and leaf yard waste, as discussed above. The model predicts that this combination of options would sequester 14,583,850 $MgCO_2eq$ by 2100, and increase of 3,826,060 $MgCO_2eq$ over the baseline scenario.

Table 16: Fairfax County Maximum and Balanced Carbon Sequestration during Transitioning to Renewable Energy

Scenario	2020-2100 Carbon Sequestered or Offset ($MgCO_2eq$)	
	Total	**Difference from Baseline**
Tree Planting		
- Balanced tree planting from 2020-2040	**11,056,866**	**+299,076**
Food Waste		
- Compost: start in 2033; 90% by 2046	**11,058,957**	**+301,167**
Herbaceous Yard Waste		
- Burn: 100% starting in 2020	**11,371,612**	**+613,822**

Leaf Yard Waste		
- Burn: 100% starting in 2020	**10,825,992**	**+68,202**
Wood Flooring Waste		
- Recycle: 100% by 2030	10,663,676	-94,114
- Landfill: 100% in 2020	**11,628,290**	**+870,500**
Lumber Waste		
- Recycle: 100% by 2033	10,819,075	+61,285
- Landfill: 100% in 2020	**13,113,107**	**+2,355,317**
Baseline Scenario	10,757,790	-
Maximized Scenario	**15,265,874**	**+4,508,084**
Balanced Scenario	**14,583,850**	**+3,826,060**

Sensitivity Analysis

This study uses data for soil organic carbon that describe the Southeastern region rather than the Mid-Atlantic region, as defined by (Pouyat et al., 2006). Fairfax County is on the border between these two regions. It is worth noting that there are substantial differences in soil organic carbon values between the regions for grassland/shrubland and forest land cover between the two regions. Pouyat et al. (2006) report that these differences arise from differences in climate and soil types; however, such differences are unlikely to occur at precisely the location of political boundaries. Therefore, this study considered how the use of the Mid-Atlantic region values might affect the model results.

Using the Mid-Atlantic region values increased soil carbon sequestration, but not sufficiently to change the results of this study. The difference in values was only $1.33 \cdot 10^{-9}$ $MgCO_2eq$ per year for 20 years after the land cover changed.

Uncertainty

Calculating carbon sequestration and emissions offsets produces estimates that carry a level of uncertainty. One source of uncertainty is in using IPCC values that are regional or national estimates. The carbon content of forest wood and tree growth rates are given by the

IPCC at a regional level and thus may carry some error when applied to a specific locale. Similarly, the IPCC methodology used in this study does not consider urban trees to sequester carbon once they reach 20 years old due to the effects of pruning and trimming, but this estimate may not apply as well in suburban cities where many residential trees may not be regularly trimmed. For example, tree canopy covers 60% of residential land in Fairfax County, with an additional 25% eligible for tree cover expansion (O'Neill-Dunne et al., 2017). Given the extensive land area involved, it seems most likely that the majority of these trees are not trimmed on a regular basis. The model developed for this study could be improved by providing for a more detailed accounting of tree carbon content, forest growth rates, and management practices for urban trees.

Another source of uncertainty or error in this study has to do with a lack of some of the data necessary to fully characterize carbon sequestration in forests. The IPCC methodology used in the model does not account for Nowak and Crane (2002)'s finding that sequestration levels change based on tree planting density. Further, while Nowak and Crane (2002) found that a forest of older, bigger, more densely spaced trees sequester up to twice as much carbon as younger, smaller, sparsely planted forests, other researchers have come to contradictory conclusions. Additional research is needed to determine more precisely how carbon sequestration changes with tree planting density as well as tree age, as mentioned above. Further, while tree planting density likely varies between urban trees and forest trees, data were not available to describe this density difference. This study therefore assumes that tree planting density is uniform across Fairfax County and any other jurisdiction to which the model is applied. The possible impact of this error is impossible to estimate given the general lack of data on tree density in forests, parks, and developed lands.

Additionally, the estimates for soil carbon sequestration used in this study assumed equilibrium values that represent average soil carbon for various land uses. It is unknown whether these values represent local conditions for the case study area, nor how well they approximate soil organic carbon for other cities to which the model in this study may be applied. The IPCC also notes the importance of collecting data on soil organic carbon levels at regular intervals in order to account for changes in equilibrium points associated with changes in local climate and land management practices (Intergovernmental Panel on Climate Change (IPCC), Aalde, et al., 2006). The generalized nature of the soil carbon values used in this study may therefore be a source of error if data from the case study area proves to differ significantly from the values used.

Finally, the factors used in this model that are based on the EPA WARM model also carry uncertainties with the potential for error. The model in this study uses the WARM model's assumption of a 20-mile transport distance for waste products (U.S. EPA, Office of Resource Conservation and Recovery & ICF International, 2019a), which may be a source of error if the actual transportation distance is much different from this value. For composting, the impact on soil carbon is based only on data for degraded agricultural soil (U.S. EPA, Office of Resource Conservation and Recovery & ICF International, 2019d); it is unclear how such soil differs from the typical soils found in cities. The EPA also notes that the N_2O emissions factors used in the WARM model and the current study are averages based on studies finding a wide range of values; therefore, the N_2O emissions factor likely introduces some error to the model results. Finally, EPA notes that the emission factors for recycling wood waste are based on studies of manufacturing practices during the 1990s. It is possible that these practices, and their associated emissions, have changed over the past 20-30 years, though the magnitude and direction of any such change is unknown.

Discussion

This study found that policies affecting the management of direct and embodied carbon sinks do affect the amount of carbon sequestered in those sinks, potentially by several orders of magnitude. For embodied carbon, the study found that the waste management practice that maximizes carbon sequestration may vary by waste product, and that the impacts of different policy options on carbon sequestration may change over time as carbon emissions change in the jurisdiction. In enabling such policy comparisons, the model developed for this study was demonstrated to be capable of providing estimates of carbon sequestration for various policy options in the context of a specific city's existing carbon sinks, management policies, and projected future carbon emissions.

For Fairfax County, the study found that the County's decision to replace landfilling of embodied carbon waste products with a policy of burning them in a WTE facility sequestered the most carbon; indeed, this policy change converted the County's embodied carbon from a source to a sink. However, the study also found that the County could further increase its carbon sequestration by burning only food waste and herbaceous and leaf yard waste, while rerouting CDD wood waste back to landfills. The model estimated that burning herbaceous yard waste would increase carbon sequestration by nearly twice as much as any other policy change. Landfilling CDD wood flooring waste was estimated to sequester more than eight times more carbon than landfilling CDD lumber waste, and nearly four times more carbon than burning leaf waste. The model also found that making all of the changes necessary to maximize sequestration of embodied carbon stored more carbon than tree planting campaigns targeting primarily vacant land and parkland.

Comparing these same policy options under a scenario in which Fairfax County transitions to 100% renewable energy by 2070, the model estimated that there is a point at which current policies affecting carbon sinks become less effective. For Fairfax County, the model estimated that the transition to renewable energy would make composting food waste a better option for increasing carbon sequestration than burning this waste product decades before the transition to renewable energy is complete. The model did find that burning herbaceous and leaf yard waste, and landfilling CDD wood waste, would continue to maximize carbon sequestration during and after the transition to renewable energy. By 2100, the model estimated that the carbon sequestration from landfilling CDD lumber and wood flooring waste would have been more than three times sequestration from burning food and yard waste. Further, the model estimated that these maximized waste management practices would result in carbon sequestration ten times greater than that achieved by a balanced tree planting campaign beginning in 2020 and completing 27 years later.

Finally, this study assessed the impact of efforts to enhance city carbon sinks. The study found that if Fairfax County achieves 100% renewable energy, annual sequestration in the County's carbon sinks under balanced changes to waste management practices plus a modest tree planting campaign would amount to 1.4% of the County's 2015 carbon emissions (Metropolitan Washington Council of Governments, 2018). Without optimizing carbon sinks, the County would sequester 1.0% of its 2015 carbon emissions each year after transitioning to renewable energy. On average, the balanced practices were the equivalent of annually taking 39,630 cars off the road, or 10,397 cars/year more than baseline management practices (based on the EPA's estimate that the average passenger car emits 4.6 $MgCO_2eq$ annually (U.S. EPA, 2016)).

It is difficult to compare these results with previous studies due to the lack of other research covering the same spectrum of urban carbon sinks; however, findings from studies of individual carbon sinks provide comparison. For direct carbon sinks, this study's results are consistent with Mohareb & Kennedy (2012)'s findings regarding sequestration in urban carbon sinks in Toronto, Canada. Mohareb & Kennedy (2012) found lower annual rates of carbon sequestration per hectare of forest, but their findings are in line with the findings in this study after adjusting for the higher rate of annual forest growth in Fairfax County, which the IPCC estimates is 3.75 times greater than in Toronto, Canada (Intergovernmental Panel on Climate Change (IPCC), 2006). Mohareb & Kennedy (2012) also found that direct sinks sequestered more carbon than embodied sinks, which is consistent with the findings in this study.

For embodied carbon sinks, both Mohareb & Kennedy (2012) and the current study found that embodied carbon sent to landfills without any CH_4 capture is a net emissions source. It is not possible to compare the findings of this study with Mohareb & Kennedy (2012)'s results for landfills with CH_4 capture as the latter study did not account for any GHG emissions from such landfills. No other previous study seems to have considered the carbon accounting associated with managing all of the embodied carbon waste products considered in this study.

The current study's results do conflict with findings from Morris (2017), but the methodology used in the two studies diverged in important respects. Morris (2017) compared different management practices for wood waste, finding that recycling sequestered more carbon than burning or landfilling. Morris' study did not consider the carbon emissions associated with harvesting virgin timber though, which the IPCC estimates as lower than emissions from the wood recycling process. Morris (2017)'s study also assumed zero fiber loss in recycling wood products, while the WARM model on which this study is based estimates that 1.25 tons of wood waste are needed to replace 1.1 tons of virgin wood (U.S. EPA, Office of Resource Conservation and Recovery & ICF International, 2019a). The current study found recycling to be a net carbon source when accounting for these differences. Further, Morris (2017) derived emission factors for fossil fuels from the average emissions of all fossil fuel power plants nationwide, while EPA values used in this study are adjusted to reflect the marginal rate of power plant emissions for specific U.S. regions ((U.S. EPA, Office of Resource Conservation and Recovery & ICF International, 2019c). Finally, Morris (2017) did not account for biogenic carbon storage in landfills, which this study considers as sequestered indefinitely.

In terms of overall carbon sequestration, Mohareb & Kennedy (2012) found that direct and urban carbon sinks in Toronto amounted to only 2% of that city's carbon emissions. This study's finding that carbon sinks in Fairfax County corresponded to 1% of the County's annual carbon emissions is in keeping with the study of Toronto. The lower value found in the current study may be to the differing characteristics of the two regions considered (e.g., there are differences in the size of direct sinks and embodied carbon waste streams between Toronto and Fairfax County). In addition, Mohareb & Kennedy (2012) considered gross carbon sequestration, while this study calculated net carbon sequestration. Mohareb & Kennedy (2012) noted that there was potential for increasing carbon sequestration in urban sinks; however, the current study found that this potential is small compared to the size of urban carbon emissions, with the maximum carbon storage found for Fairfax County amounting to carbon sequestration of, on average, 1.4% of the County's annual emissions.

Several limitations to the current study suggest that the findings for overall carbon sequestration may be an underestimate, however. The most basic limitation is that this study did not model all possible urban carbon sinks. Sewage sludge is one potential embodied carbon sink that was not included in this study due to a lack of data on its emissions or sequestration factors. Cement was also excluded from this study on the assumption that emissions from its manufacturing process may prevent cement from acting as a carbon sink across its full lifecycle. However, recent research suggests that a methodology for zero emissions production of cement may be on the horizon (Ellis, Badel, Chiang, Park, & Chiang, 2019). Both sludge and cement thus represent possible embodied carbon sinks.

Another limitation of the current study was the inability to account for certain aspects of the carbon sinks that were modeled. One factor this study was unable to account for is deadwood removals from forests, the impact of which is unknown. Removing deadwood from forests reduces carbon storage in the forest, according to the IPCC carbon accounting methodology (Intergovernmental Panel on Climate Change (IPCC), 2006). Orozco-Aguilar, Johnstone, Livesley, & Brack (2018) found that this deadwood can contain 0.069-0.110 MgCO2eq/tree, but this was based on a study of only three species, none of which are found in the case study area. Mohareb & Kennedy (2012) note that while accounting for deadwood reduces the estimated value of carbon sequestration in forests, the overall impact may be much reduced if the removed deadwood is converted to wood products that are ultimately disposed of through processes that sequester their embodied carbon. The impact of not considering deadwood in the case study is therefore expected to be negligible.

This study also did not account for recent research suggesting that increased diversity of tree species in a forest may increase the forest's carbon sequestration. A study in China found that each additional tree species in a forest increased carbon sequestration by 6.4% (Liu et al., 2018). Accounting for the possibility of increased carbon sequestration due to tree species diversity in the model developed for the current study would require additional studies of the impacts of tree species diversity in other regions, as well as how the rate of increase in carbon sequestration, if any, varies as the number of tree species changes.

Similarly, the accuracy of this study's results would be improved by further research on the emissions from landscaping activities that may have been necessary on vegetated land prior to its conversion to tree cover as modeled in this study. Fairfax County reports that lawn and garden equipment emissions are included in the County's greenhouse gas inventory as part of a broader category of "off-road" equipment emissions used in the County. The net emissions from all equipment in this category amounted to 3.7% of the County's total carbon emissions in 2006 (Bulova et al., 2013). Depending on the contribution of lawn and garden equipment to this total, the emissions from their use could cancel much or even all of the carbon sequestration provided by Fairfax County sinks prior to any transition to renewable energy sources. This study therefore considers the emissions from lawn and garden equipment to be an important area for future research.

Finally, the application of this study's results is limited by the fact that the study did not consider the full range of environmental impacts – much less the broader economic or political impacts – associated with each policy option for managing carbon sinks. For example, this study found that landfilling wood waste maximized sequestration of embodied carbon; however, Morris found that landfilling wood waste increased emissions of air pollutants that increase the risk of respiratory disease in humans, while the other management options offset these pollutants (Morris, 2017). Morris also found that recycling wood products reduced emissions of chemicals thought to increase the risk of cancer in humans, eutrophication, acid rain, and overall ecosystem toxicity. Similarly, this study found that composting food waste maximized carbon sequestration during a transition to renewable energy; however, compost has a low economic value so a city may need to enact laws that require residents to recycle food waste and charge them a fee for the additional waste collection service in order to make the process of composting economically viable (Meyer-Kohlstock, Hädrich, Bidlingmaier, & Kraft, 2013). Conversely, this study found that burning rather than composting yard waste results in greater carbon sequestration, but generating compost and applying it to the soil may increase plant productivity, reduce erosion, and increase soil nutrients (U.S. EPA, Office of Resource Conservation and Recovery & ICF International, 2019d). Finally, this study does not account for the broader environmental impacts that may be associated with increases in virgin timber harvest that occur when wood waste is not recycled. These impacts may have be broad depending on size of the increase in virgin tree harvest, and include possible losses of soil carbon, losses other soil nutrients, and increased erosion, as suggested by Van Hook, Johnson, West, & Mann (1982), and possibly decreased wildlife habitat or even decreases in biodiversity.

Similarly, for direct urban sinks, this study found that tree planting campaigns measurably increase carbon sequestration, but such campaigns are not without drawbacks. The U.S. Forest Service found that afforestation may be a high-cost and high-maintenance venture, with campaign sites requiring intensive watering and hand-pulling of unwanted plants for several years (Hallett, 2013). Further, afforestation and even reforestation of vegetated land may displace habitat for arthropods and other insects (Threlfall & Kendal, 2018). The model developed in this study focuses solely on carbon accounting, but does not consider the broader ecological or other impacts of policies for managing urban carbon sinks.

Conclusion

Overall, this study finds that changes in policies for managing city carbon sinks can have a small but measurable impact on carbon sequestration. This is consistent with the previous study of urban carbon sinks by Mohareb & Kennedy (2012). This study refines Mohareb & Kennedy (2012)'s process of estimating carbon sequestration in urban sinks by calculating net rather than gross carbon sequestration. The uncertainties in the model developed for this study, and the limitations on modeling the totality of urban carbon sinks, raise the possibility that the carbon sequestration estimates provided in this study may underestimate the possible sequestration in urban areas.

The model developed in this study provides a tool for comparing the relative impacts on carbon sequestration of different policy options for managing urban carbon sinks, in the context of size of an individual city's direct carbon sinks and embodied carbon waste streams, the city's existing policies for managing these direct and embodied sinks, and the city's projected future carbon emissions. The model may support policymakers' abilities to identify how carbon sequestration changes during a transition to renewable energy, and the waste management policy changes that could optimize carbon sequestration before, during, and after such a transition. Due to the exclusive focus on carbon sequestration, the model results should be considered in the context of the broader environmental, economic, and health impacts that may be associated with policies for managing direct carbon sinks and embodied carbon products in urban waste streams.

Appendix: Model Screenshots

The figures below provide screenshots of the modules representing direct and urban carbon sinks.

Figure 2: Forest Trees Module

Figure 3: Urban Trees Module

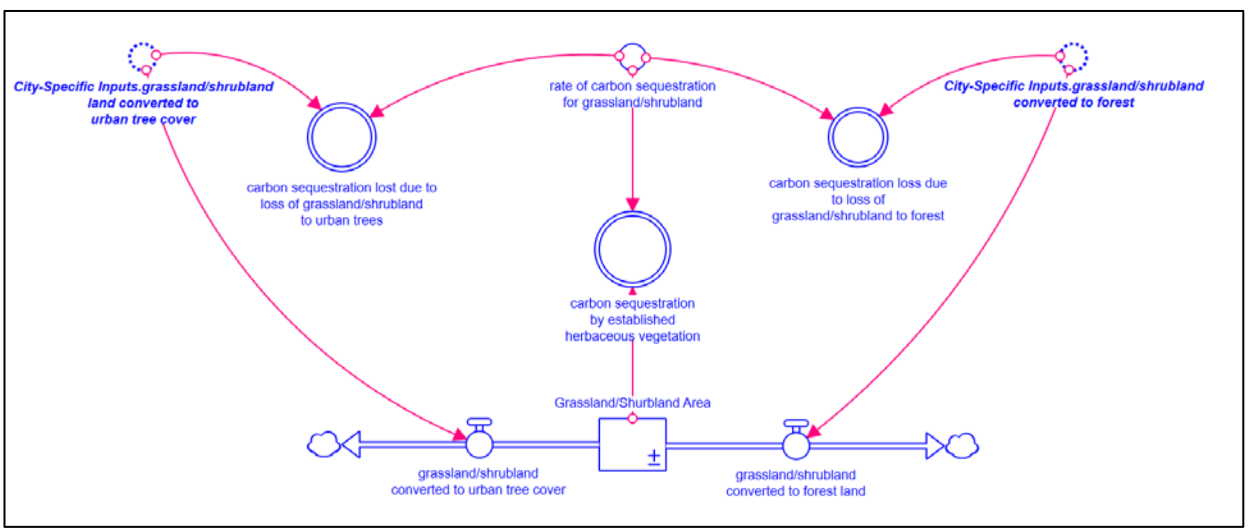

Figure 4: Herbaceous Vegetation Module

Figure 5: Soil Carbon Module

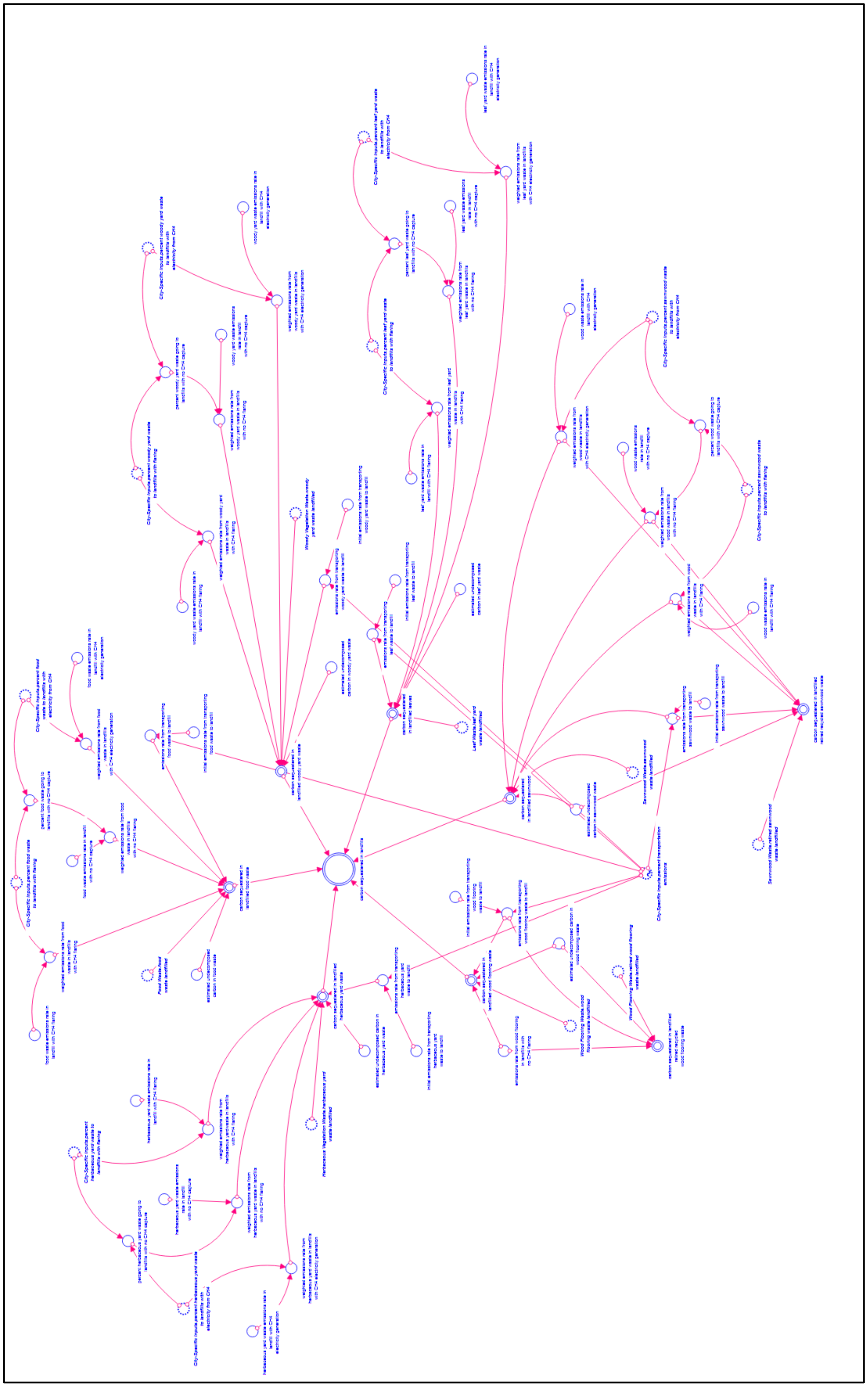

Figure 6:Landfilling Module

Figure 7: Composting Module

Figure 8: Recycling Module

Figure 9:Combustion Module

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